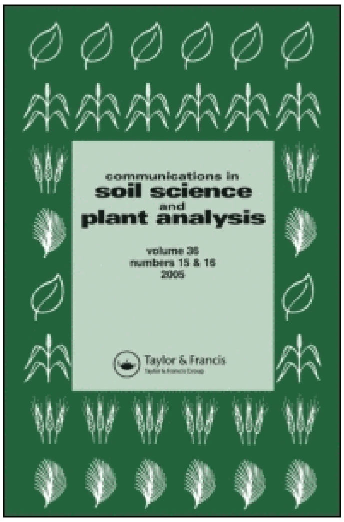

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Safeguarding Soil and Water Quality

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ABSTRACT

In many countries, community awareness and concern about environmental issues has resulted in an increased interest in and requirement for assessment and monitoring of soil and water quality. This paper reviews the types of indicators which are commonly advocated for these purposes, and concludes that better interpretive guidelines for indicators are needed if these guidelines are to be defensible. More work also needs to be done to decrease the cost of appropriate monitoring and to encourage its wider and more intense use. Recent developments in new techniques, and community-based monitoring programmes in Australia are discussed. Active and well targeted assessment and monitoring programmes by themselves are not sufficient to safeguard soil and water quality. Increased emphasis needs to be given to the development of new land use systems and practices, which address the environmental priorities identified through assessment and monitoring, and which also ensure the financial viability of land managers. Scientists have important roles to play in all of the above processes. Research is one important role, but there is much to be gained from scientists facilitating communication between land managers and policy makers. These gains include the advancement of reforms to land management which have the potential to safeguard soil and water quality, and which are likely to be adopted by land managers and local communities.

INTRODUCTION

Advances in our understanding of the impacts of human activity on soil and water resources continue to be made. We now know that certain land management practices can decrease the capability of land or water for continued use *in situ* or the capability of water for alternate use elsewhere (Sumner and McLaughlin, 1996; Lovering and Crabb, 1998). Many of the concerns relate to the accumulation of plant nutrients as a consequence of over fertilising, and their transport to other parts of the landscape where they may have undesirable effects (White and Sharpley, 1996; Sharpley and Rekolainen, 1997; Carpenter et al., 1998). Other concerns include sediment in water from rural and urban land disturbance (CAST, 1992; USEPA, 1994; Sumner and McLaughlin, 1996), and accumulation of heavy metals and agricultural pesticides in both soil and water (White and Kookana, 1998; McLaughlin et al., this Symposium; Kookana and Simpson, this Symposium).

The purpose of this paper is not to discuss the above issues in detail. We start from the viewpoint that in many regions soil and water quality is already compromised or under threat. Increasingly, community expectations are that governments will act to improve the management of these resources while still meeting the needs of present users. In response, many governments throughout the world have adopted policies for sustainable development and for monitoring the state of the environment, and have also legislated to better regulate the use of land and water (e.g. USEPA, 1993; New Zealand Ministry for the Environment (MfE), 1996; State of the Environment Advisory Council, 1996; Clement and Bennett, 1998; Parry, 1998; USDA-USEPA, 1999). In Australia, laws regulating water use and quality are currently under review in most states (see Clement and Bennett, 1998; Department of Land and Water Conservation, 1998; DPIF Tasmania, 1998). Legislation is also being developed at state and national levels in the United States of America (USA) to safeguard soil and water resources, while maintaining the economic viability of agricultural production and rural infrastructures (Harkin, 1997; Simpson, 1998; USDA-USEPA, 1999).

Under such legislation, there is an increasing requirement for soil and water quality to be assessed against defined standards, although the implementation of such assessments and the definition of standards can lag behind the legislation (MfE, 1996; Parry, 1998; Sparling and Rijkse, 1998). In some instances, these laws are pushing the limits of the science upon which they depend. This can place pressures on land and water users by making them comply with standards that may be imprecisely defined or open to challenge. It also heightens challenges for science to bridge the gap between theory and practice in assessing the state of land and water resources, in delivering affordable tools for this purpose and in devising appropriate and affordable resource management systems for the future.

The purpose of this paper is to briefly review current views, mainly in Australia, the USA and New Zealand, on indicators to best assess soil and water quality. Ways in which community and legislative expectations about safeguarding soil and water quality might be fulfilled are then discussed. The role that science and scientists can play in helping the community and government to fulfill these expectations is also discussed.

MATERIALS AND METHODS

What to Measure?

Environmental indicators have been defined as "physical, chemical, biological or socio-economic measures that can be used to assess natural resources and environmental quality" (State of the Environment Advisory Council, 1996). There has been much discussion about which characteristics of soil and water are the most appropriate indicators of the current state (so-called condition indicators) and rates of change (trend indicators) in the quality of those resources. The New Zealand Ministry of Environment (1997) describes a good indicator as one which is analytically valid, cost effective, simple and easily understood, and relevant to current environmental policy and legislation. In Australia, Dalal et al. (1998) have developed a set of ten selection criteria for indicators for the Queensland grains industry (Table 1) which are based on those proposed earlier by Walker and Reuter (1996). The concept of selection criteria helps accommodate the fact that no single indicator will have universal application nor is without its limitations.

TABLE 1. Selection criteria for deciding on suitable indicators of soil and water quality (from Dalal et al., 1998 after Walker and Reuter, 1996).

Criterion	Possible scoring system
Responsiveness to change in management or disturbance over time	0 = non-responsive 10 = responsive
Ease of capture	0 = difficult, needs specialist training 10 = easy, even for non-specialist
Interpretation criteria available	0 = not available 10 = universally available
Error associated with measurement	0 = high error 10 = low error
Stable during period of measurement	0 = extreme fluctuation 10 = very stable
Required frequency of measurement	0 = very frequent 10 = infrequent
Cost	0 = high cost (>AU\$100 ha^{-1}) 10 = low cost (<AU\$10 ha^{-1})
Ability for aggregation from paddock to farm to catchment to regional scale	0 = difficult 10 = easy
Capacity for mapping in space and time	0 = no capacity 10 = high capacity
Community acceptance and involvement	0 = none 10 = full acceptance

It is apparent from Table 1 that the selection of indicators of soil and water quality can be quite subjective because scoring many of the selection criteria requires judgement. Nevertheless, when lists of recommended soil and water quality indicators are compared, there is much consistency from list to list in the types of measures which are selected (Tables 2-5). For example, common soil measures include pH, nutrient status and organic carbon. This is not surprising because these tests are routinely used to assess soil fertility. Soil physical measures vary somewhat, but generally aim to describe how well air and water enter and are stored in the soil, and whether plant root growth will be affected. Some soil biological measures are advocated in all but the New Zealand system. However, there is a recognition that at the moment many of these tests are not yet robust (Pankhurst et al., 1997; Dalal, 1998) since they lack basic interpretive criteria to cater for temporal and spatial heterogeneity (Doran and Parkin, 1994; Duxbury and Nkambule 1994).

Soil chemical measures have been routinely used for many years to evaluate soil fertility and to recommend the need for fertilisers and other soil ameliorants (Daubeny, 1845; Kamprath and Watson, 1980; Peverill et al. 1999). These have been particularly well developed for phosphorus (P), where extensive field trials have been conducted to calibrate various extracts of "plant available" P. These soil P tests are regionally customised according to dominant soil chemical characteristics e.g. parent material, texture, organic matter content, and Fe, Al, and Ca content (Fixen and Grove, 1990; Sharpley et al., 1994; Moody and Bolland, 1999).

TABLE 2. Indicators of resource maintenance for the central Queensland grains industry as suggested by Dalal et al., (1998).

Soil (on farm)	Water (off farm)
pH	Stream flow
Electrical conductivity	Stream turbidity
Organic matter	Stream water pH
Plant available nutrients	Stream water electrical conductivity
Microbial biomass	Stream water nutrient concentrations
Surface cover during fallow	Stream water pesticide concentrations
Runoff	Riparian vegetation
Erosion	Sediment
Rooting depth	Instream macro-invertebrate populations
Surface crusting	

TABLE 3. Proposed Stage 1* indicators for land and fresh water in New Zealand (MfE, 1997).

Soil or land	Water
Soil intactness (are soils staying in place?)	*Lowland slow flowing rivers*
Land use relative to capability	Dissolved oxygen
Land use	Water clarity
Land cover	Water temperature
Extent and frequency of land slipping	Ammonia concentration
Extent and frequency of water erosion	*Fast flowing rivers*
Soil health (of the soil in situ)	Water clarity
Land use relative to capability	Water temperature
Soil pH	Macro-invertebrate index
Organic carbon	*Lakes*
Soil bulk density	Dissolved oxygen
Nutrient budgets and fertiliser use	Water clarity
Nutrient status	Trophic index

*Indicators able to be used or in use now as opposed to Stage 2 indicators which will be implemented later.

For water, turbidity or water clarity is commonly advocated as an indicator of quality (Tables 2-4) because turbidity affects the amount of light reaching the aquatic ecosystem and is itself affected by both rural and urban land disturbance (MfE, 1997). It can also be readily and cheaply assessed using simple equipment (Kruger and Lubczenko, 1994) Indicators of water eutrophication are also generally suggested but the recommended indicator varies. For example, Dalal et al. (1998) suggest direct measures of nutrient concentrations (Table 2), while the New Zealand approach (MfE, 1997) measures dissolved oxygen, which can respond to high nutrient

TABLE 4. Key indicators of catchment health proposed by Walker and Reuter (1996).

Indicators of condition (the state of the system)	**Indicators of trend** (how the system is changing)
Soil consistence	Bare soil %
Soil texture	Effective root depth
Soil colour	Soil pH
Water intake rate	Soil electrical conductivity
Soil strength	Weeds %
Slaking and dispersion	Stream pH
Cotton strip test	Stream electrical conductivity
Tree cover	Stream turbidity
Groundwater EC	Macro-invertebrates
Chemical fertility	Watertable depth

TABLE 5. Basic indicators of soil quality proposed by Doran et al. (1994).

Physical	**Chemical**	**Biological**
Soil texture	Total organic C and N	Microbial biomass C and N
Depth of soil and rooting	pH	Potentially mineralisable N
Bulk density and infiltration	Electrical conductivity	Soil respiration
Water holding capacity	Mineral N, P, K	Biomass C / Total organic C
Water retention characteristics		Respiration / biomass ratio
Water content of soil		
Soil temperature		

concentrations (Table 3). However, water temperature and salinity, and diurnal variation in oxygen supply and demand due to the photosynthesis and respiration of algae and aquatic plants, all need to be taken into account when assessing dissolved oxygen concentrations (ANZECC, 1992). The inclusion of ammonia in Table 3 arises from concern in New Zealand about nitrogen (N) movement to streams from dairy pastures and effluent, and the consequent threat of a direct toxicity of ammonia to aquatic life (MfE, 1997). New Zealand is also developing a protocol for assessing algal growth. In that country, algal growth is preferred as a water quality indicator over direct measures of stream nutrient concentrations because often these concentrations are decreased through absorption by fast growing algae and aquatic plants (MfE, 1997). The challenge is to monitor change in algal growth so that action can be taken before these populations threaten other aquatic life.

Stream salinity (electrical conductivity) is an important water quality indicator in many parts of south west and south east Australia, where increasing salinity is considered to be a major threat (State of the Environment Advisory Council, 1996; Williamson et al., 1997; SCARM, 1998). However, the total salt load (volume x concentration) rather than concentration alone

TABLE 6. Threshold soil test P values and P management recommendations in the U.S. (from Sharpley et al., 1996).

State	Threshold values, mg kg^{-1}		Soil P test method	Management recommendations for water quality protection
	Agronomic[†]	Environmental		
Arkansas	50	150	Mehlich 3	*At or above 150 mg kg^{-1} soil P:* Apply no more P, provide buffers next to streams, overseed pastures with legumes to aid P removal, and provide constant soil cover to minimize erosion.
Delaware	25	50	Mehlich 1	*Above 50 mg kg^{-1} soil P:* Apply no more P until soil P is significantly reduced.
Idaho	12	50 & 100	Olsen	*Sandy soils - above 50 mg kg^{-1} soil P* *Silt loam soils - above 100 mg kg^{-1}soil P* Apply no more P until soil P is significantly reduced.
Ohio	40	150	Bray 1	*Above 150 mg kg^{-1} soil P:* Reduce erosion and reduce or eliminate P additions.
Oklahoma	30	130	Mehlich 3	*30 to 130 mg kg^{-1} soil P:* Half P rate on >8% slopes. *130 to 200 mg kg^{-1} soil P:* Half P rate and reduce surface runoff and erosion. *Above 200 mg kg^{-1} soil P:* P rate not to exceed crop removal.
Michigan	40	75	Bray 1	*Below 75 mg kg^{-1} soil P:* P application not to exceed crop removal. *Above 75 mg kg^{-1} soil P:* Apply no P from any source.
Texas	44	200	Texas A&M	*Above 200 mg kg^{-1} soil P:* P addition not to exceed crop removal
Wisconsin	20	75	Bray 1	*Below 75 mg kg^{-1} soil P:* Rotate to P demanding crops and reduce P additions. *Above 75 mg kg^{-1} soil P:* Discontinue P applications.

[†]Agronomic threshold concentrations are average values for non-vegetable crops; actual values vary with soil and crop type. Also, vegetables have higher agronomic P requirements.

better reflects the status of stream salinity (Jolly et al., 1996). Groundwater salinity may be an even better indicator because groundwater is the source of much stream and soil salinity.

Environmental concerns in the USA have forced many state and federal agencies to consider adopting standard soil P fertility tests as indicators of the potential for P release from soil and its transport in runoff. Table 6 gives examples of proposed environmental threshold concentrations from several states, along with agronomic threshold concentrations for comparison. Environmental threshold levels range from 2 (Michigan) to 4 (Texas) times agronomic thresholds. In most cases, agencies proposing these thresholds plan to adopt a single threshold value for all regions under their jurisdiction. However, threshold soil P levels are too limited to be the sole criterion to guide manure management and P applications. For example, adjacent fields having similar soil test P levels but differing susceptibilities to surface runoff and erosion, due to contrasting topography and management, should not have similar soil P thresholds or management recommendations (Sharpley, 1995; Pote et al., 1996). Therefore, environmental thresholds for soil P will have little value unless they are used in conjunction with estimates of site-specific potential for surface runoff and erosion. Soil type, and its effect on P retention, will also affect soil P thresholds. Future standards will need to account for regional variation in soil type, emphasising the need for their local calibration.

In both Australia and New Zealand, assessment of aquatic macroinvertebrates (Rosenberg and Resh, 1993; Johnson, 1995) has been advocated as an indicator of water quality (Tables 2-4), but in New Zealand assessment is restricted to fast flowing rivers, where presumably the impacts of vegetation clearance, intensive agriculture, grazing and urban settlement are less. All of these factors may greatly affect the habitat for macro-invertebrates and make more difficult comparisons of slow-flowing monitoring sites against reference sites, which are usually in "pristine" locations (National River Processes and Management Program, 1994).

In the USA, the Environmental Protection Authority (USEPA) has developed rapid bioassessment protocols (RBPs) (Plafkin et al., 1989) to detect impairments to aquatic life and for assessing their relative severity. However, once an impairment is detected, additional chemical and biological tests are needed to identify the causative agent, its source, and to implement appropriate remedial strategies (USEPA, 1991). The protocols have been recently updated to provide more cost-effective and scientifically valid approaches (Barbour et al., 1997). They now focus on an analysis of periphyton (Rodgers et al., 1979), benthic macroinvertebrates (Southerland and Stribling, 1995), and fish assemblages (Karr et al., 1997) because each of these has several advantageous characteristics, which are outlined in Table 7. Barbour et al. (1997) further suggest that the revised RBPs can be applied to a wider range of planning and management purposes than originally envisioned. For example, they may be applied to priority setting; evaluating point and nonpoint pollution sources; land use suitability analyses; and trend monitoring, as well as initial screening.

It is worth re-emphasising here that the choice of both soil and water quality indicators will vary between regions and farming systems. Those monitoring soil and water quality have to decide what they need to measure and how often it should be done. For example, there is little point spending money measuring soil electrical conductivity in well drained, high rainfall regions or soil erosion on permanent pastures growing on flat landscapes. The choice of indicator may also depend on the scale at which information about that indicator is to be reported and used. For example, measures of stream salinity or groundwater depth have relevance in regional or catchment scale salinity assessments for identifying where drainage infrastructure should be installed. However, at the farm scale it would be better to measure soil electrical conductivity to identify paddocks where salt tolerant species should be planted.

Interpretation of Indicators

Soil Quality Indicators

In Australia and New Zealand, criteria for interpreting soil chemical and physical indicators have not yet been widely promoted, let alone included in legislation. Walker and Reuter (1996) have suggested ranges of values which define good (no problem), fair (action may

TABLE 7. Advantages of using different assemblages in aquatic biosurveys (from Barbour et al., 1997).

Assemblage characteristic	Advantages
Periphyton (mainly algae)	
Rapid reproduction rates	Valuable indicators of short-term impacts
Primary producers	Directly affected by physical and chemical factors
Sensitive to pollutants	Other aquatic life may not be visibly affected or at high pollutant levels only
Benthic Macroinvertebrates	
Limited migration	Indicate local conditions
Complex life cycles	Integrate effects of short-term environmental variations
Wide range of trophic levels	Good indicators of cumulative effects
Sampling relative simple	Little detrimental effect on resident biota
Fish	
Long-lived and mobile	Indicate long-term effects and broad habitat conditions
Many species and trophic levels	Integrate lower tropic level effects and environmental health
Top of aquatic food web	Able to assess contamination and are consumed by humans
Easy to collect	Fish are half of the endangered vertebrate species and subspecies in U.S.

be needed to address or further investigate a problem), poor (action definitely needed) or very poor (urgent action needed) categories for Australian indicators. These guidelines are based on previous research but are necessarily general and not site specific. Further Australian interpretive criteria for both soil physical and especially chemical attributes are given in Peverill et al. (1999), although these are mostly for agricultural production and not for environmental protection. Criteria are also under evaluation in New Zealand (Sparling and Rijkse, 1998). In both countries, there is recognition that in most cases appropriate criteria will be specific to particular environments (land use, climate, soil type), so that local knowledge needs to be applied to any general guidelines. One way to link this approach to legislation is to have local communities set the standards for appropriate land management practices, as is the case in South Australia where regional Soil Conservation Boards formed under state legislation prescribe their own standards in an approved district plan. Adoption of a similar process is being debated in the USA. Local people are in the best position to modify general guidelines as appropriate. However, any modifications should be validated by scientists for radical departures from acceptable practice. To the extent that soil and water quality indicators become measures required to gain access to markets, the choice and interpretation of such indicators must also be acceptable to those markets.

Several states in the USA have attempted to adopt threshold soil P levels (Table 6), to limit the land application of P, particularly as manures, biosolids, and other by-products. In all cases, the legislation was repealed due to legal challenges against these soil thresholds, because the legislation directly related the thresholds to water quality degradation in a technically indefensible way. New nutrient management legislation in various stages of development (for example, in Maryland, Delaware, and Virginia) will state that threshold values will be based on the best science available and on soil-water relationships being developed (Simpson, 1998; Lander et al., 1998). This course is also followed in the joint USEPA-USDA strategy for sustainable nutrient management for animal feeding operations (AFOs) (USDA-USEPA, 1999). This draft strategy proposes a variety of voluntary and regulatory approaches, whereby all AFOs would develop and implement comprehensive nutrient management plans by the year 2008. These plans deal with manure handling and storage, application of manure to the land, record keeping, feed management, integration with other conservation measures, and other options for manure utilisation. The draft strategy is out for public comment, and will be revised and in place by the end of 2001 for poultry and swine operations and by 2002 for cattle and dairy enterprises. This leaves scientists only 2 to 3 years to develop "the best science available" that includes technically defensible thresholds or indicators.

Water Quality Indicators

Australian guidelines for environmental water quality are given in ANZECC (1992) but these do not set rigid limits within which all water quality parameters should lie. For pH and turbidity, they instead define the degree of change which indicates a possible disturbance to the ecosystem and which should therefore trigger further investigation (Jolly et al., 1996). For some indicators, the Australian guidelines in ANZECC (1992) have recently been reviewed alongside those from individual Australian States (Liston and Maher, 1997). Separate thresholds are proposed for indicators depending on the altitude of the water body (Table 8). Guideline total P concentrations are less stringent for lower (0.05 mg P/L) than for higher altitude waters (0.02 mg P/L) because suspended solids in lowland waters can decrease the biological activity of P. The guidelines for total P (Table 8) are consistent with those set by the USEPA (0.05 mg P/L for streams entering lakes and reservoirs and 0.025 mg P/L within lakes and reservoirs (USEPA, 1988), but lower than those used in the Netherlands (0.15 mg P/L; Van der Molen et al., 1998).

In the USA, states are required to set their own water quality criteria, but so far only 22 states have quantitative standards and only Florida has adopted the federal USEPA levels (Parry, 1998). These standards include designated water uses, water quality criteria to protect these uses, and an anti-degradation policy. Where water quality standards are not attained, even where best management practices have been implemented, response actions are defined through the Total Maximum Daily Load (TMDL) process of the 1998 Clean Water Action Plan (USEPA, 1998). This process not only addresses constituent concentrations in stream and rivers, but also considers system discharge and thereby the total constituent load, as well as the designated use and potential impact on the receiving water body.

Integrating Indicators of Soil and Water Quality

Indices of overall soil quality, obtained by arithmetic weighting and combination of scores for individual indicators, have been proposed (e.g. Doran et al., 1994). The weightings are usually subjectively assigned, and the indices usually combine quite different measures in an empirical rather than mechanistic way. Walker et al. (1996) do not favour this approach, preferring that the importance of each indicator is individually evaluated. Nevertheless, there is often a desire to make an overall statement about any multi-faceted assessment. As long as any combining of information does not mean that one or more problem issues are overlooked, indices can be a useful way to package a "take-home message". So far there seems to have been only limited practical testing of such indices. Dalal et al. (1998) describe an adaptation of a multiple-

TABLE 8. Recommended guidelines for water quality in Australia (from Liston and Maher, 1997).

Indicator	Altitude		
	<100 m	100-500 m	>500 m
Total P (mg L^{-1})	50	50	20
Total N : Total P	15:1	15:1	15:1
BOD (mg L^{-1})	10	5	5
Turbidity (NTU)	20	15	10
Suspended solids (mg L^{-1})	40	30	20
Chlorophyll (μg L^{-1})	20	20	10

objective decision support system, originally developed in the USA to compare different resource management systems, which they used to assess the relative importance of sustainability indicators. The system can be used at both regional and local levels. It seems sensible for indices to be developed at the local level so that the weightings applied to individual indicators are acceptable for that area.

One approach to a soil and water quality index in the USA integrates soil fertility measures (specifically soil test P; Table 9), land management and a site's potential to transport nutrients to water bodies in surface and subsurface runoff. This approach is being advocated by researchers and an increasing number of advisory personnel, to address nutrient management and the risk of nutrient transport at multi-field or catchment scales (Lander et al., 1998; Maryland General Assembly, 1998; USDA-USEPA, 1999). In cooperation with research scientists, the USDA Natural Resource Conservation Service has developed simple nutrient indices as screening tools for use by field staff, catchment planners, and farmers to rank the vulnerability of fields as sources of N and P loss (Sharpley et al., 1998). The indices account for and rank transport and source factors controlling N and P loss in subsurface and surface runoff and identify sites where the risk of nutrient movement is expected to be higher than others. These indices have been incorporated into state and national nutrient management planning strategies which address the impacts of animal feeding operations on water quality. They help identify agricultural areas or practices that have the greatest potential to pollute water resources (Simpson, 1998; USDA-USEPA, 1999).

Inherent differences exist in the geography and biology of different agricultural regions, forests, wetlands, and water bodies (Bailey, 1976; Energy, Mines and Resources Canada, 1986). As water bodies reflect the lands they drain (Hunsaker and Levine, 1995), an ecoregional framework that describes similar patterns of naturally occurring biotic assemblages, such as land-surface form, soil, existing natural vegetation, and land use was proposed by Omernik (1987) and later refined by USEPA (1996b). The ecoregion concept provides a geographic framework for efficiently managing aquatic ecosystems and their components (Hughes, 1985; Hughes et al., 1986; Hughes and Larsen, 1988). For example, studies in Ohio (Larsen et al., 1986), Arkansas (Rohm et al., 1987), and Oregon (Hughes et al., 1987; Whittier et al., 1988) have shown that distributional patterns of fish communities approximate ecoregional boundaries as defined *a priori* by Omernik (1987). This, in turn, implies that similar water quality standards, criteria, and monitoring strategies are likely to be valid within a given ecoregion (USEPA, 1996b). Eco-

TABLE 9. Indicators of landscape vulnerability to N and P loss in subsurface and surface runoff (from Sharpley et al., 1998).

Transport Factors	Source Factors
Rainfall intensity and duration	Soil N and P content
Runoff class Soil texture Soil permeability	Fertilizer N and P - • Rate • Method • Timing
Erosion potential Landscape position Site proximity to stream channel	Manure N and P - • Rate • Method • Timing

regional boundaries for Australia have been defined as 80 biogeographical (Thackway and Cresswell, 1995) or 11 agro-ecological regions (SCARM, 1993; 1998). Both regional types have been used to analyse regional trends in environmental quality.

RESULTS AND DISCUSSION

Making Monitoring More Affordable and Widespread

One of the main challenges facing those monitoring soil and water quality is meeting the costs involved. This challenge applies not only to the farmer wanting to characterise his own land, but also to the policy maker seeking to draw conclusions at a catchment, regional or national scale. At the moment, the tools available appear to be conventional analysis of soil and water samples, which are relatively expensive at the intensity of sampling required. Reuter (1998) has reviewed a number of developments which offer the potential to make such assessment more affordable. These include:

1. Estimate plant yield and water use efficiency (after French and Schultze, 1984) as an integrative surrogate for agricultural soil quality to at least partially avoid the need for expensive soil analyses. This has greatest potential in lower rainfall cropping systems.
2. Global positioning systems and other technologies of precision agriculture (Cook and Bramley, 1998) may yield sampling strategies which are more efficient than conventional approaches, or perhaps more likely, will lead to the development of automated samplers which may reduce costs (Viscarra Rossel and McBratney, 1998).
3. The availability of field kits and meters for testing various attributes of both soil and water *in-situ* is increasing, and offers the opportunity for land managers to cut costs provided they have the time to do the tests themselves (see Kruger and Lubczenko, 1994). Soil quality kits are now available from several USA companies and nonprofit sustainable agricultural groups. Progress is also being made towards field based analytical systems which employ sensors capable of directly assessing various soil chemical attributes (Viscarra Rossel and McBratney, 1998; Ridings this Symposium) which may avoid the need for sampling, often the most expensive part of a monitoring programme.

4. In the laboratory, the use of mid infrared spectroscopy has potential to supplant more expensive and time consuming physical and chemical tests with minimal sample preparation (Janik et al., 1998), although the scope for mimicking some traditional soil nutrient extractions seems limited. Ultimately, this technology may be packaged in field portable form.

5. In Australia, more serious consideration needs to be given to the use of multi-element or universal extractants as a means of decreasing analytical costs. These extractants have been used in the USA for many years (Jones, 1990) but have yet to gain widespread acceptance in Australia (Reuter, 1998).

Whatever technology is used to assess soil and water quality, an additional universal requirement will be locating monitoring sites so that data can be spatially integrated for assessment at larger scales. Meeting this requirement is being made increasingly easy by the falling price of mobile global positioning systems. Georeferencing is probably more a requirement for scientists and regulators who are monitoring soil and water quality at regional or catchment scales. Individual land managers could well be satisfied relating their data to particular paddocks or other land features.

A critical component in making monitoring more widespread is the funding of cost-share programmes and the development of alliances among stakeholders. In the USA, there are numerous sources of technical assistance and financial cost-share and loan programmes to help defray the costs of constructing or implementing practices that safeguard soil and water resources. Some of these sources are Conservation Technical Assistance (CTA), Conservation Reserve Programme (CRP), Environmental Quality Incentives Programme (EQIP), Special Water Quality Incentives (SWQI), Wetlands Reserve Programme (WRP), and Wildlife Habitat Incentive Programme (WHIP) (USEPA, 1998). In 1998, CRP spent approximately US$500 million to establish an estimated 172,000 miles of buffers throughout the USA. EQIP was funded at US$200 million in 1997 and 1998. In each of these years, requests for funds were about 3 times that available and USDA has requested US$300 million for EQIP in 1999.

Stakeholder alliances encourage collaborative relationships among the groups involved. Such alliances have been formed in response to recent public health issues related to the nutrient enrichment of waters in the eastern USA. Several outbreaks of the dinoflagellate *Pfiesteria piscicida* in the eastern USA over the last five years, have been linked to high nutrient levels in affected waters (USEPA, 1996a; Burkholder and Glasgow, 1997). Neurological damage in people exposed to the highly toxic, volatile chemical produced by this dinoflagellate has dramatically increased public awareness of eutrophication and the need for solutions (Matuszak et al., 1997). The dinoflagellate outbreaks have had a major economic impact in several areas. Both the fishing and tourist industries have been severely affected in the Chesapeake Bay and inland waters of the Delmarva Peninsula and North Carolina. For example, fish sales in Maryland decreased by up to 75% in 1997, and a marketing campaign to reassure customers that fish from Chesapeake Bay were safe to eat was funded at $200,000 (Meyer, 1997). In North Carolina waters, an estimated 1 billion fish are suspected to have been killed over the past ten years by dinoflagellate outbreaks. Overall, the economic loss to the affected eastern coastal states is estimated at $1 billion over the last two decades (Greer, 1997). In the Chesapeake Bay, stakeholder alliances have developed among state, federal, and local groups and the public to work together to identify critical problems, focus resources, include catchment goals in planning, and implement effective strategies to safeguard soil and water resources (Chesapeake Bay Program, 1995; 1998).

Another way of making some of these environmental programmes more affordable is to increase public awareness and involvement. For example, a multi-agency collaborative venture called the Dairy Network Partnership, has just released Chesapeake Milk in the Northeast USA. For every half-gallon sold, 2.5 cents will be returned to certified Pennsylvania dairy farmers to reward their high environmental standards. Another 2.5 cents will be deposited into an Environmental Quality Initiative (EQI) fund which will provide a cost-share for those farmers who want to install conservation practices to qualify for the EQI programme.

Purpose of Monitoring: Finding Pressure Points and Doing Something About Them

Increasing community concern about the environment has caused governments to spend more on monitoring activities. Recent initiatives in state of the environment reporting and environmental auditing in Australia (State of the Environment Advisory Council, 1996; National Land and Water Resources Audit (NLWRA), 1998; SCARM, 1998) attempt to put environmental assessment and monitoring on a more rigorous, consistent and repeatable platform so that knowledge of the main environmental issues and how they should be tackled steadily improves. However, much is already known about the problems Australia faces with its soil and water resources, so in addition to finding out more about these problems, an equally pressing need is to design and implement new and profitable management systems which ensure these resources are adequately protected for the future. As Wylie (1994) puts it, "Monitoring is mostly a watching it happen occupation instead of doing something about it". An awareness of problems is necessary before one can then act to correct them, but awareness of problems and even having appropriate solutions does not necessarily cause people to change their behaviour to correct such problems (Wilkinson and Cary, 1993). Solutions have to be adapted in practical ways to individual circumstances. The Australian National Land and Water Resources Audit has recognised this in its strategic plan by including investigations of the capacity of rural communities to implement change needed to help protect soil, water and vegetation (NLWRA, 1998).

One example where an issue has progressed from awareness to action comes from the USA, where an awareness of soil and water quality concerns associated with manure and nutrient management in animal feeding operations, created a dialogue within the pork and poultry industries to address these concerns. From this dialogue, USEPA and the US National Pork Producers Council (NPPC) produced in late 1998 a voluntary environmental compliance programme to safeguard soil and water resources and public health from the wastes of pork-producing operations (NPPC, 1997; USEPA and NPPC, 1998). The NPPC has been proactive in developing a self-management programme in cooperation with federal agencies to monitor and assess pork-producing facilities throughout the USA. As the pork producers have a vested interest in the success of this programme, it is expected that practical solutions will be developed and implemented.

One barrier to the design and implementation of farm management systems which protect soil and water resources is that the assessment and monitoring implemented by government is often perceived as a top-down process. In contrast, Walker et al. (1996) recommended a bottom-up process where soil and water quality indicators are selected by local communities. A bottom-up process would seem equally if not more important to the successful identification and adoption of new management systems. A concerted attempt has been made in Australia to take this approach by devolving responsibility for local monitoring and resource management to land managers themselves through the provision of government funds to the National Landcare and Waterwatch programs. The national vision for the Decade of Landcare Plan is "the development and implementation of systems of land use and management which will sustain individual and community benefits now and in the future" (SCARM-ARMCANZ, 1997). Over 4000 Landcare groups were operating in 1997 (Landcare Australia, 1997), and the total allocation of funds in 1994/5 was AU$103m (Alexander, 1995). Groups usually provide half the costs of their projects, although much of their contribution is "in-kind". About a third of broadacre and dairy farms in 1995/6 were represented on Landcare groups (Mues et al., 1998).

The mission statement for Waterwatch in Australia is "to promote water quality monitoring as a means of creating and enhancing ownership ethic for broad scale environmental management by the Australian people" (Commonwealth of Australia, 1994). Waterwatch has recently been incorporated into a reorganised National Landcare Program (SCARM-ARMCANZ, 1997). Many Landcare groups are involved in water quality monitoring, and schools have also formed Waterwatch groups since the programme has a particular emphasis on training and education (Commonwealth of Australia, 1994). Gowland (1997) reported that there were at least 1100 Waterwatch groups in Australia in December 1994.

There is evidence that Landcare does encourage monitoring of land and water resources. Alexander (1995) reports that 71% of South Australian groups and 60% of Tasmanian groups who responded to a survey were involved in monitoring changes to their physical environment. Furthermore, a survey by the Australian Bureau of Agriculture and Resource Economics (ABARE) found that, compared to non-members, members of Landcare groups in relevant farming zones were more likely to monitor watertable depth and test their soils (Mues et al., 1998). Members of Landcare groups were also more likely to have reported that they had problems of soil and water degradation, and to have adopted practices such as monitoring the condition of pastures, maintaining riparian zones and practicing conservation tillage. However, many of the differences between Landcare members and non-members were relatively small, and other factors such as whether a respondent had completed some form of related training or education were also important in explaining adoption patterns.

Whether Landcare promotes or merely reflects an interest in resource stewardship amongst its members has not been established. A view has also been expressed that Landcare funding tends to centre activities at the regional not the farm level (Williams, 1995), which is supported by data of Macgregor and Pilgrim (1998) who found that about 40% of the funds spent in Western Australia through the National Soil Conservation Program/National Landcare Program were for professional assistance to groups and another 20% for land resource assessment and mapping. Macgregor and Pilgrim (1998) highlight the risk of top-down decision making with such an emphasis on professional assistance, but they also indicate that many farmer groups appreciate the initiative and organisational and technical skills that such professional support provides.

The fact that training and education appear influential in accelerating the adoption of improved practices, and that both are important parts of both Landcare and Waterwatch, suggests that these programmes may provide increasing benefits in the future. Within some sections of Landcare, training and education are strategies preferred over government regulation and incentives (Alexander, 1995), but there is also a counter view that a lack of government incentives in Australia has hampered the adoption of good management practices (Williams, 1995). The same author also considers that the broadening of Landcare to encompass most aspects of natural resource management has diluted the funding available for its "original" focus of agricultural land management (Williams, 1995). Waterwatch has not been evaluated in the same way as Landcare.

How Science Can Assist in Safeguarding Soil and Water Quality

The preceding discussion has highlighted a number of areas where science can and must assist communities to safeguard soil and water quality. These goals, together with possible roles for scientists in seeing that these goals are reached, are presented in Table 10. In many areas, research is a priority strategy, with perhaps the most fundamental issue being research into better understanding regional landscape processes, particularly regional biogeochemistry and water balances (Reuter 1998). This will provide a foundation for continued research into appropriate

TABLE 10. Future opportunities for science to assist in safeguarding soil and water quality.

Outcome 1	**Appropriate indicators used to assess and monitor soil and water quality**
Strategies	• Validate indicators and understand their limitations before promoting them • Help the community prioritise soil and water quality issues • Develop protocols to assist community to select indicators appropriate for their needs
Outcome 2	**Monitoring is affordable and widespread**
Strategies	• Continue to develop simple, low-cost methodologies including remote sensing techniques • Develop reliable kits that place appropriate tools for monitoring in the hands of land managers
Outcome 3	**Community has realistic targets for soil and water quality**
Strategies	• Understand landscape processes - develop and validate models • Develop and refine techniques for integrating point source information to catchment or regional scale • Assess limitations of current strategies • Develop defensible interpretive criteria • Bridge the gap between land managers and policy makers
Outcome 4	**Land and water management practices safeguard soil and water quality**
Strategies	• Understand social and economic constraints to change • Work from the bottom up with land managers to refine existing and develop new practices that maintain physical and economic farm viability • Help land managers to make sense of existing information and to resolve conflicting information • Help the community prioritise soil and water quality issues

new and refined indicators, the assessment and validation of those indicators, and research into new or better ways of managing the land profitably for the long term.

While research will continue to be an important role for scientists, Table 10 also highlights a major on-going communication role. This will be important in helping to bridge the gap between land managers and policy makers. The role is partly one of interpreter. On the one hand, scientists can help convey to land managers the reasons why community expectations are driving policy makers to change the regulations about land and water management. On the other, we can identify and explain characteristics of land use systems and their economics to regulators so that the best policies are put in place. In both cases, our knowledge will be important in making sense of existing information, much of which is conflicting (Wylie, 1994; Sharpley and Tunney, 1999), and in realistically assessing the importance of soil and water issues alongside other important community issues.

In being information brokers between land managers and regulators, scientists can also help by giving more emphasis to the bottom-up flow of information than has previously been the case. The role here is to identify practices needed to meet specific soil or water management goals, to present them to the community, and to refine them where necessary so that they are accepted and adopted by the community.

Through this communication role it is likely that many opportunities for research will arise, opportunities which will have a better chance of being realised because government and particularly industry ownership has been engendered through the communication process. In

Australia in the last decade, industry-based rural Research and Development Corporations have been directing an increasing amount of rural research funds. More recently, regional farmer groups such as Southern Farming Systems in Victoria, Tasmania and South Australia, have initiated their own research programs and field sites. These trends heighten the need for scientists to understand issues from industry's point of view.

A good example of the gap between policy and science is being vigorously debated in the U.S. In new legislation and required implementation of nutrient management plans, particularly for agricultural systems involving animals, there has been a shift in focus from N to P (Lander et al., 1998; Sharpley et al., 1998). This shift has been driven by an increased incidence of freshwater eutrophication and toxic dinoflagellate outbreaks (USEPA, 1996a; Burkholder and Glasgow, 1997; Matuszak et al., 1997), even though the link between N or P and the dinoflagellate outbreaks has not been established. Because of this shift from N to P, some in the farming community feel misled by science and extension, which recommended N-based manure management to mitigate nitrate leaching to ground water (Blankenship, 1997; Matuszeski, 1997; Achenbach, 1998). Associated soil P build up was encouraged to enhance soil fertility. This policy was fuelled by several misconceptions, especially that soil is an infinite sink for P and that erosion control will eliminate P loss from agricultural fields (Sharpley, 1997). Knowledgeable scientists must become more proactive in disseminating the best information to the diverse community of soil and water resource users. However, because scientists work with statistical probabilities and long-term complex interactions, their time frames for delivering the "best" information are often longer than those of government and the community, which seek more immediate solutions.

An example of the research which might flow from involvement in industry-government communication is farm scale research to evaluate all external and internal factors controlling nutrient balances and export to water bodies. Farmers are at the start of the "food chain" and their decisions are increasingly influenced by regional and often global economic pressures and constraints, over which they or their industries can have little or no control. In the U.S. dairy industry since World War II, greater fertiliser N use, which increased corn grain production and reduced costs, along with the promotion of a domestic soybean processing industry, has markedly increased the feed energy and protein available for enhanced animal productivity (Lanyon, 1999). Improved animal breeding, specialized feed concentrates, and new production technologies promoted greater animal productivity on a smaller land area. Animal farming has changed from land-based to capital or economically-driven systems. External economic factors and not environmental considerations therefore increasingly drive issues like manure production and management.

Computer Simulations

Science has attempted to simulate the effect of changing land and water management practices through the use of computer models. These have the potential to integrate information on a large scale, and to locate within a catchment specific high impact areas which need remediation. For example, the AGNPS (Agricultural Nonpoint Pollution Source, Young et al., 1989 and 1995) model was developed to provide estimates of the quality of runoff water from agricultural catchments of up to 50,000 acres and to assess the effects of applying remedial management measures to targeted areas. The model operates on a cell basis that makes it possible to analyse discrete management units (fields) within a catchment and thereby identify fields that have a relatively greater potential to be a source of exported nutrients. Managing nonpoint sources of nutrients from large catchments is otherwise complex because the origins of these sources can not always be readily identified (Gburek and Sharpley, 1998).

More recently, a conceptual, continuous time model called SWAT (Soil and Water Assessment Tool) was developed to assist water resource managers in assessing the impact of land management on water supplies and nonpoint source pollution in catchments and large river basins (Arnold et al., 1998). The model is currently being utilized in several large scale U.S. projects to estimate the off-site impacts of climate and management on water use, nonpoint source loadings, and pesticide contamination. For more detailed information on other models in this field, reviews by Leavesley et al. (1990) and Rose et al. (1990) are available. A directory of relevant Australian models which describes their aims, nature and intended users, has recently been compiled by Hook (1997).

While these models have much to offer, they are still largely research tools (Wylie 1994), and while they may help farmers to understand their farming system, they are no substitute for on-farm implementation of management changes.

CONCLUSIONS

Safeguarding soil and water quality requires multidisciplinary research teams working with land managers and rural communities. This work should include the development of a more complete understanding of landscape processes (i.e. sources, sinks and pathways), and of practical management options to halt or reverse soil and water degradation. For any given region, the processes and priority options will vary.

Rather than conclude *prima facie* that inappropriate farm management is responsible for today's soil and water quality problems, we must address the underlying causes of the symptoms. In many cases, these causes are related to marketplace pressures and the economic survival tactics of farmers. Policy support and financial incentives will be needed to change land management practices in many areas, and to provide solutions which are acceptable to communities.

To ensure monitoring programmes are widely adopted, future research also needs to focus on developing new and technically smarter indicators of soil and water quality. This research includes the development and use of low-cost laboratory, field and remote sensing technologies.

Emphasis must also be given to defining defensible criteria and thresholds for soil and water quality assessment which can be applied reliably to local environments. Such criteria are essential if community misperceptions about proper environmental management are not to become reality.

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